Chapter 17

Contingent Valuation of Forest Ecosystem Protection

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In recent decades, concerns have arisen about the proper valuation of the world's forests. While some of these concerns have to do with market distortions for timber products or inadequate data on non-timber forest products, an additional challenge is to uncover the economic worth of non-market services provided by forest ecosystems (Kramer et al. 1997). This has led to a growing number of publications addressing the valuation of forest ecosystem services, on topics such as carbon sequestration and endangered species habitat. In this chapter, we focus on the contingent valuation method (CVM) to assess the structure, health, and extent of forest ecosystems.[1]

Forest ecosystems generate a wide variety of use values, the most important of which are timber, non-timber products, recreation, wildlife habitat, and watershed services. While use values are important, and their provision was the primary objective of public land management in the past, increasingly public land managers are confronted with demands arising from passive use values such as the knowledge that specific ecosystems exist or will be available for future generations to enjoy. Although use and passive use values are both non-market, passive use values can only be measured using stated preference methods.

Researchers use one of two stated preference methods, CVM and Attribute Based Method (ABM), to uncover non-market values of forest quality. ABMs represent a merging of the hedonic method in economics that views the demand for goods as derived from the demand for attributes, with marketing research methods for determining perceived values of particular product features. Values are revealed through a series of questions that ask people to rate, rank, or choose among a set of alternatives with varying levels of each attribute. ABMs have been used in recent years for valuing

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particular attributes of forests, e.g., age, species variety, and watershed protection. The use of ABMs is described in more detail elsewhere in this volume (see chapter 18). CVM remains an important tool for forest resource economists because forest ecosystems present bundles of goods and services that cannot be easily separated. In fact the components of a forest ecosystem often move together. For example, a forest with greater levels of species diversity may also have higher levels of watershed services and aesthetic value than less diverse forests. Thus, one can think of contingent valuation as a tool that is appropriate for valuing complex environmental goods such as forest ecosystems precisely because it leads to a holistic approach rather than focusing on individual components. Also, contingent valuation is appropriate for valuing unique resources. Estimating economic values for forest ecosystems can improve the formation and implementation of policies to manage those ecosystems.

In this chapter, we first review studies that used contingent valuation to evaluate forest quality, health, and extent. We find strong evidence that forest ecosystem condition can be considered an economic good and is therefore a candidate for cost/benefit analysis of forest protection actions. We then present a case study on forest quality in the Southern Appalachian Mountains. The spruce-fir ecosystem in this area is undergoing rapid change due to environmental stress. Although spruce-fir forests in the Southern Appalachians currently provide little in the way of commercial or market amenities, they provide significant non-market values, including recreation, scenic beauty, and biodiversity protection. We present results from an earlier study of this ecosystem and a new analysis of the consistency of measured willingness to pay (*WTP*) values along with a discussion of the theoretical constructs that allow an economic interpretation of measured forest values.

1. LITERATURE REVIEW

1.1 Existence Values/Passive Use Values for Forests

Existence value—what people are willing to pay to protect resources they have no plans to use—has emerged as the most important non-use or passive use value associated with environmental resources. Krutilla's widely cited paper "Conservation Reconsidered," provided the first formal argument for including existence value in benefit estimates: "When the existence of a grand scenic wonder or a unique and fragile ecosystem is involved, its preservation and continued availability are a significant part of the real income of many individuals" (Krutilla, 1967:779). In a footnote, Krutilla stated that "These would be the spiritual descendents of John Muir, the present members of the Sierra Club, the Wilderness Society, National Wildlife Federation, Audubon Society and others to whom the loss of a species or the disfigurement of a scenic area causes acute distress and a sense of genuine relative impoverishment."

Some non-economists have raised ethical objections to monetizing existence values of resources (Adams 1990), but the resource economics profession has pushed ahead with its valuation agenda, arguing that failure to do so will result in significant under-valuation of environmental resources in policy and management decision making. A variety of motivations for existence value have been proposed in the literature (Boyle and Bishop 1985, Brookshire et al. 1986, Krutilla 1967, McConnell 1997, Randall and Stoll 1983). The motivations include several types of altruism as well as bequest motives. In addition, there is vicarious consumption derived from reading books or watching documentaries about nature. Although this vicarious consumption could be seen as indirect use, in practice it cannot be separated from pure existence value (Smith and Desvousges 1986). While Boyle and Bishop (1985) consider sympathy for other species and concerns about environmental linkages as part of altruistic behavior, McConnell (1992:3) argues that preference for the natural order is distinct from altruism, which he defines as *WTP* to preserve a resource because the resource "enhances the well-being of others." We view it as plausible that some forest areas would meet with Krutilla's criteria regarding existence values for "unique and fragile ecosystems." While there is a lack of consensus in the literature about exactly what types of preferences are represented by existence values, if existence values enter into the total value of an ecosystem (either singly or in combination with use values), it is generally agreed that their influence should not render total value estimates inconsistent with economic theory. Recently, the argument has been made that well-behaved preferences for existence goods can be evaluated by examining the consistency of measured values with signs of the first and second derivatives of the *WTP* function (Diamond 1996, Loomis and Larson 1994, Rollins and Lyke 1998). Complete specification of a consistency test is a major focus of this chapter, and is discussed in section 2 below.

1.1.1 Previous Non-market Forest Valuation Studies

The first studies using CVM to estimate values for forest protection appeared in the early 1990s and were generally concerned with forest degradation due to insect infestations and air pollution (table 17.1). Walsh et al. (1990) used the iterative bidding technique to estimate the value of protecting ponderosa pine on national forests in the front range of the Colorado Rocky Mountains. This study confirmed that the general public is willing to pay for forest protection programs. In addition, by asking respondents to decompose total value into four categories of value (recreational use, option, existence, and bequest), the authors concluded that use values accounted for 27.4% of total value, and non-use values (including option, bequest, and existence values) accounted for 72.6% of total value.

Table 17.1. Contingent valuation studies of forest protection

Author	Year	Type of experiment [a]	Type of activity	*WTP* value
Haefele, Kramer, and Holmes	1991	PC, DC	Protect high-elevation spruce-fir forest in southern Appalachian mountains from exotic insect and air pollution	PC = $21/yr. DC = $100/ yr.
Jakus and Smith	1991	DC	Increase in aesthetic quality of homeowner property due to gypsy moth; private use value	$238–$394 for private control; $295–$494 for a public control
Kramer and Mercer	1997	PC, DBDC	Creating national parks and protected areas to preserve 10% of tropical rain forests	PC = $31/ yr. DBDC = $21/ yr.
Li and Mattson	1995	DC	Continued access to the forest environment under the Swedish Right of Common Access	12,817 Swedish Kroner ($1,600), adjusted for preference uncertainty
Loomis, Gonzalez-Caban, and Gregory	1996	OE, DC	Reduce fire hazard to old-growth forests	OE = $33/yr. DC = $98/yr.
Miller and Lindsey	1993	DC	Protect homeowner property from gypsy moth by state-run control program; private-use value	$69/ yr.
Reaves, Kramer, and Holmes	1999	OE, PC, DBDC	Restore 75,000 acres of old-growth longleaf pine for red-cockaded woodpecker habitat	OE = $11/yr. PC = $8/yr. DBDC = $13/yr.
Walsh, Bjonback, Aiken, and Rosenthal	1990	Iterative bidding	Protect mixed-age ponderosa pine from mountain pine beetle	$47/ yr.

[a] PC = payment card, DC = dichotomous choice, DBDC = double-bounded dichotomous choice, OE = open ended

Haefele et al. (1991) reported a positive *WTP* for protecting high-elevation spruce-fir forests in the southern Appalachian Mountains from exotic insect infestation and air pollution damage. Using a decomposition approach, they found that non-use values (bequest and existence values) dominated use values as reasons for protecting these forests. Subsequent analysis of responses given by people who never had visited the study area and did not intend to visit the study area in the future confirmed that ecosystem existence values were substantial and empirically distinct from total ecosystem values (Holmes and Kramer 1996).

Jakus and Smith (1991) and Miller and Lindsay (1993) used CVM to estimate *WTP* for gypsy moth protection programs. These studies differed from the earlier studies in that private, not public, property was the focus of valuation.

Loomis et al. (1996) elicited preferences of Oregon households for reducing fire hazards to old-growth forests in the Pacific Northwest. They also used two CVM response formats and found that *WTP* estimated from dichotomous choice responses was greater than *WTP* estimated from open-ended responses, which is consistent with the Holmes and Kramer (1995) study. It is interesting to note that dichotomous choice estimates for protecting old-growth forests from fire ($98/year) are very close to dichotomous choice estimates for protecting spruce-fir forests from insect epidemics/air pollution damage ($100/year). Likewise, payment card estimates of protecting spruce-fir forests ($21/year) are similar to open-ended estimates of *WTP* to protect old-growth forests ($33/year). Kramer and Mercer (1997) evaluated the preferences of a random sample of U.S. citizens regarding the creation of protected areas to preserve 10% of tropical rain forests. In contrast to Holmes and Kramer (1995), they found that *WTP* computed from dichotomous choice responses were lower than *WTP* estimated from payment card responses. Payment card estimates of *WTP* for creation of rain forest preserves ($31/year) were similar to payment card estimates of protecting spruce-fir forests ($21/year) and open-ended estimates of *WTP* to protect old-growth forests ($33/year). The dichotomous choice estimates of *WTP* for creation of rain forest preserves were, in general, lower than *WTP* values estimated using the dichotomous choice method in other studies.

Finally, one study estimated the value of restoring old-growth longleaf pine forests in South Carolina (Reaves et al. 1999). These forests were severely damaged by a natural event (hurricane) and provided habitat for an endangered species (the red-cockaded woodpecker). *WTP* estimates were quite similar for three valuation methods used. In addition, restoration *WTP* values were somewhat lower for this resource than for protection activities in old-growth forests in the Pacific Northwest, spruce-fir forests in the Southern Appalachian Mountains, and tropical rain forests.

2. CONCEPTUAL MODEL OF CVM CONSISTENCY

A major focus of this chapter is to evaluate the consistency of values measured using the CVM with constructs of neoclassical economic theory. We begin our discussion with the proposition that consumer preferences for the condition of a forest ecosystem can be represented by a utility function. Neoclassical economic theory states that utility is quasi-concave with respect to quantity or, equivalently, that preferences (indifference curves) are convex with respect to the origin (e.g., see Johansson 1987).[2] Thus, the first increment in quantity of an economic good should have a positive value. A second increment in quantity should also have a positive value (non-satiation), but the increase in value should be less than the first increment (diminishing marginal value).

This proposition implies that people are willing to make substitutions among bundles with varying levels of market goods and forest conditions. Some authors have argued that goods that embody existence values, such as endangered wildlife species, may invoke lexicographic preferences based on ethical concerns (Edwards 1986, Edwards 1992, Stevens et al. 1991). The lexicographic rule always ranks one characteristic of a decision problem above another. In the present context, a lexicographic decision rule would always rank improvements to ecosystem condition above other considerations, such as changes in household expenditures. That is, a household with lexicographic preferences would never be indifferent between various combinations of forest ecosystem conditions and expenditures on other goods and services. Lexicographic preferences are not well-behaved from an economic perspective.

If forest condition can be represented by a well-behaved utility function, then increments in the forest area protected will increase utility at a diminishing rate. A measure of the economic value of an increment in forest condition is the amount of money an individual is willing to pay to attain the increment and which leaves the individual just as well off as if there were no increment in forest protection and no payment. This measure is known as the compensating surplus and can be written using the expenditure function, which minimizes household expenditure subject to the constraint that utility equals or exceeds some reference level (Freeman 1993). In particular, the compensating surplus is written as the difference between two expenditure functions:

$$Compensating\ surplus = e(p, q^0(a^0), u^0) - e(p, q^1(a^1), u^0) \tag{17.1}$$

where p is a vector of market prices; q is a measure of forest condition, which, in turn, is a function of the area protected a; and u is utility. The superscript 0 refers to the *status quo*, and the superscript 1 refers to the changed condition. The compensating surplus is a welfare theoretic measure of *WTP* for a specific increment to forest ecosystem condition.

A positive *WTP* for an initial increment in forest condition beyond the status quo suggests that forest ecosystem condition can be considered an economic good and is therefore a candidate for cost/benefit analysis of forest protection actions. The first hypothesis to be tested is whether *WTP* for an initial increment is statistically different than zero. The null hypothesis is:

$$H_0^1 : WTP_a = 0 \tag{17.2}$$

where a is a measure of the area protected. The null is tested against the alternative hypothesis that $WTP_a > 0$.

Second, if preferences for forest ecosystem condition are consistent with consumer theory, then people will be willing to pay more for greater levels of protection. This suggests a second null hypothesis: people gain utility from protecting a core area of a forest ecosystem, but do not gain marginal utility from protecting more than the core area. To test this hypothesis, we establish the null hypothesis that incremental *WTP* for incremental gains in forest condition is equal to zero:

$$H_0^2 : \int_{q^0}^{q^1} \Theta(p, q(a), u^0)\, dq = 0 \tag{17.3}$$

where marginal *WTP*, $\Theta(p,q(a),u^0)$, is a partial derivative of the expenditure function:

$$\Theta(p, q(a), u^0) = \frac{-\partial e(p, q^1(a^1), u^0)}{\partial q} = \frac{\partial WTP}{\partial q} \tag{17.4}$$

The partial derivative represents the slope of the individual's indifference curve at the point of evaluation, and marginal *WTP* is integrated over the incremental change $q^0(a^0) \rightarrow q^1(a^1)$. Failure to reject the second hypothesis would imply that indifference curves are flat with respect to changes in forest protection—a violation of the neoclassical assumption regarding non-satiation. Letting the increment in the area of forest protection be represented by *b*, the second hypothesis can be rewritten:

$$H_0^2 : WTP_{a+b} = WTP_a \qquad 17.5$$

If the second null hypothesis is rejected, and the alternative hypothesis that marginal *WTP* is positive is accepted, then consistency of measured *WTP* with economic theory requires that the second derivative of *WTP* with respect to area protected be negative. The *WTP* curvature condition can be evaluated by comparing the average slope of two segments of the *WTP* function with respect to the forest area protected. In particular, the third null hypothesis is:

$$H_0^3 : \frac{WTP_{a+b} - WTP_a}{b} = \frac{WTP_a}{a} \qquad 17.6$$

In equation 17.6, the numerator in each expression represents incremental *WTP*, where it is implicitly assumed that *WTP* for no protection is zero, and the denominator represents the change in forest area protected. If the third null hypothesis is rejected, and the alternative hypothesis is accepted (the second derivative of *WTP* with respect to area protected is negative) then consumer preferences regarding forest ecosystem protection are consistent with the constructs of economic theory.

3. EXPERIMENTAL SETTING

Our experiment focuses on protection of the high-elevation spruce-fir forest ecosystem in the Southern Appalachian Mountains. This ecosystem covers 26,610 ha of mountaintops and high ridges in Virginia, North Carolina, and Tennessee. About three-fourths of this ecosystem is located in the Great Smoky Mountains National Park. This park receives about 9 million visitors per year and is the most heavily visited national park in the country.

Since the 1950s, there has been a dramatic increase in spruce-fir mortality in this ecosystem. Using aerial photography, a recent inventory determined that in one-fourth of this area, greater than 70% of the standing trees were dead (Dull et al. 1988). Research also indicates a decline in the growth rate of red spruce (*Picea rubens* Sarg.) on Mt. Mitchell, the highest mountain east of the Mississippi River (Bruck 1988). Decline of the spruce-fir forest is highly visible from roads and trails. The cause of decline of Fraser fir (*Abies fraseri* Poir.) is generally attributed to the balsam woolly adelgid, an exotic forest pest accidentally introduced from Europe. Also, some scientists have attributed the decline of these forests to air pollution impacts, through direct impacts on soils and foliage and indirect impacts on susceptibility to insect attacks (Hain 1987).

For our experiment, we considered the reference level of utility to be associated with the status quo forest condition. Because the entire ecosystem was at risk of degradation, reference utility was associated with protecting none of the existing forest area. Then, the first increment of forest protection was specified to occur along road and trail corridors, spanning one-third of the entire ecosystem at risk. This level of protection may be particularly appealing to people who value the ecosystem principally for recreational use. The second level of protection was for the entire ecosystem. It was thought that this level of protection may be appealing to people who value the ecosystem as a whole and may focus attention on the continued existence of the entire threatened ecosystem.

4. SURVEY INSTRUMENT

A contingent valuation mail-out mail-back survey was used to gather information about *WTP* for protection of the remaining healthy spruce-fir forests, along with information about socio-economic and other characteristics of the respondents. The format of the survey and its implementation closely followed the Dillman (1978) method. The sampling frame was people living within a 500-mile radius (approximate one day's drive) of Asheville, North Carolina. This sampling frame was used because we wanted a large share of our respondents to have some familiarity with the study area prior to receiving the questionnaire.

A sheet of color photographs representing three stages of forest decline and a map identifying the study area were included with the survey along with information about forest damage and forest protection programs. Two *WTP* response formats were used: payment card and dichotomous choice. A comparison of *WTP* models and estimates from the two response formats is reported elsewhere (Holmes and Kramer 1995). In this chapter, we only use responses to the dichotomous choice questions.

Response rate to the single-bounded dichotomous choice version of the questionnaire was 51% and resulted in 221 usable observations. Of those people responding to the questionnaire, 4% did not respond to the dichotomous choice questions.

Two sequential dichotomous choice questions were posed. The first question provided information to test the first hypothesis—that people have a positive *WTP* (WTP_a) for forest ecosystem protection—and asked whether or not people would be willing to pay a specified annual amount in higher taxes to protect spruce-fir forests along roads and trails (about one-third of the remaining forest area). The second question provided information to test the second and third hypotheses—incremental *WTP* (WTP_{a+b}) increases at a decreasing rate—and asked whether or not people would be willing to pay a specified annual amount in higher taxes to protect the entire ecosystem.

Specific dollar amounts were randomly assigned across questionnaires. Identical amounts were used in both questions within a questionnaire. This assignment method was used because if dollar amount in the second question exceeded the dollar amount in the initial question, then respondents may have construed that an increasing incremental value was being sought. Further, a decreasing amount in the follow-up question may have been construed as illogical.

Four response patterns to the dichotomous choice *WTP* questions were observable: No-No (NN), No-Yes (NY), Yes-No (YN), and Yes-Yes (YY). The hypothesis test that *WTP* increases at a decreasing rate critically depends on the pattern of NY responses. A NY response would indicate that *WTP* for protecting forests only along road and trail corridors was less than the bid amount *$X*, but that *WTP* equalled or exceeded *$X* for protecting the entire forest ecosystem. Other responses would indicate either a constant or decreasing *WTP* as the area protected increased.

Respondents who answered yes to the second *WTP* question were asked a follow-up question to provide information about their *WTP* rationale. In particular, people were asked to decompose their total *WTP* into four categories, by percentage: (1) use of forests for myself, (2) use of forests for others (including future generations), (3) protection of the forests even if no one uses them, and (4) other. This question was designed to identify the importance of non-use values associated with forest ecosystem protection.

5. EMPIRICAL METHODS

Sequential presentation of *WTP* questions in our experiment suggests that responses to these questions are not independent if unobserved factors influence both responses. Single-equation models of *WTP* should not be used in cases where equation errors are correlated, because preference parameter estimates are inefficient, and standard errors of the preference parameters are upwardly biased (Greene 1997). In turn, this bias affects hypothesis testing, because standard errors of *WTP* values, and differences in *WTP* values, are computed based on parameter estimates. To obviate these problems, we used a bivariate probit model to estimate preference parameters and identify correlation in the unobserved factors influencing responses across the two *WTP* equations. In general, a bivariate probit model is specified as:

$$y_1^* = \beta_1' x_1 + \varepsilon_1, y_1^* = 1 \, if \; y_1^* > 0, \; 0 \; otherwise$$
$$y_2^* = \beta_2' x_2 + \varepsilon_2, y_2^* = 1 \, if \; y_2^* > 0, \; 0 \, otherwise$$
$$E[\varepsilon_1] = E[\varepsilon_2] = 0 \qquad 17.7$$
$$Var[\varepsilon_1] = Var[\varepsilon_2] = 1$$
$$Cov[\varepsilon_1, \varepsilon_2] = \rho$$

where y_j is the response to *WTP* question j, the β_j's are vectors of preference parameters, the x_j's are vectors of explanatory variables, and the ε_j's are the equation errors. To simplify the interpretation of the model results, we use the same set of explanatory variables in both equations ($x_1 = x_2$).

WTP values should be nonnegative for economic goods. A non-negativity constraint can be imposed by assuming that the relationship between WTP_i and $\beta_j'X_i$ is log-linear. Median *WTP* is computed from the log-linear estimates as $\exp(\beta_j' x_j/\mu)$, where μ is the parameter estimate on the bid amount Hanemann and Kanninen 1999:327. Mean *WTP* computed from a log-linear specification includes a term for the estimated variance of the model's error $(1/\mu^2)$: $WTP_{mean} = WTP_{median} \bullet [\exp(1/\mu^2)]$. Thus, model specification errors can directly lead to inflated WTP_{mean} values with the log-normal model (Huang and Smith 1998). Further, because statistical tests using median *WTP* have greater statistical power than tests based on means (Mitchell and Carson 1989, Kealy and Turner 1993), we chose to use WTP_{median} in the tests below. For example, our second and third hypothesis tests were conducted using incremental *WTP* computed as $\exp(\beta_1' x_1) - \exp(\beta_2' x_2)$.

Hypothesis tests were conducted using the Krinsky-Robb (1986) bootstrap technique. This technique is used more often than the traditional bootstrap technique in estimating *WTP* confidence intervals because of its relative efficiency. This is because the traditional bootstrap resamples the raw data, and the model must be re-estimated for each draw (Efron and Tibshirani 1993). In contrast, the Krinsky-Robb procedure uses random draws from estimation results. [3]

The first hypothesis, $WTP_1 = 0$, was tested using the estimation results from the first equation in the bivariate probit model and the achieved significance level (ASL), which is defined as "the probability of observing at least that large of a value when the null hypothesis is true" (Efron and Tibshirani 1993:203). The ASL using the bootstrap percentile method ($ASL_{\%}$) for the first hypothesis test is written as:

$$ASL_{\%} = \Pr(WTP_a = 0) = \frac{\#(WTP_a^{median} < 0)}{B} \qquad 17.8$$

where # is the number of times the condition is true and B is the number of bootstrap replications. The ratio on the right-hand side of equation 17.8 is a percentage indicating the significance level of the test.

The second hypothesis, $WTP_{a+b} = WTP_a$, was tested using the ASL:

$$ASL_{\%} = \Pr(WTP_{a+b} = WTP_a) = \frac{\#(WTP_{a+b}^{median} < WTP_a^{median})}{B} \tag{17.9}$$

This test was based on the results of both equations in the bivariate probit model. In this case, the distributions of median WTP values were not sorted before conducting the test.[4]

The expression for the third hypothesis, marginal WTP does not diminish as protected forest acres increase, can be written in a simplified form using the relationship specified in our experiment regarding protected forest area: $b = 2a$ (where a = area along road and trail corridors and b = the remaining area). Substituting this relationship into equation 17.6 and simplifying yields the third hypothesis test using the $ASL_{\%}$:

$$\begin{aligned} ASL_{\%} &= \Pr((WTP_{3a(=a+b)} - WTP_a) < WTP_a) \\ &= \frac{\#((WTP_{3a}^{median} - WTP_a^{median}) > 2(WTP_a))}{B}. \end{aligned} \tag{17.10}$$

6. RESULTS

Descriptive statistics for the variables used in the empirical analysis are shown in table 17.2. Offer amounts were based on results from an open-ended WTP question in a pre-test survey. Based on pre-test results, it was decided to use an approximate log-normal offer distribution that ranged from \$2 to \$500. Data on household income were obtained using a categorical variable representing a range of incomes. People were asked if they belonged to any environmental organization or gave money to any environmental organizations or causes (no = 0, yes = 1). People were also asked to indicate how important various reasons were to them to protect the Southern Appalachian spruce-fir forests. Response categories were (1) not important, (2) somewhat important, and (3) very important. These variables were coded as 1 for the very important category and 0 otherwise.

Table 17.2. Descriptive statistics of model variables

Variable name	Description	Mean	Standard deviation
Ln_bid	Natural logarithm of offer amount	3.83 (antilog = 46.06)	1.38
Ln_inc	Natural logarithm of household income	10.40 (antilog = 32,860)	0.77
Enviro	Member of an environmental organization	0.29 (dummy variable)	0.45
Rec_val	Recreational opportunities very important	0.38 (dummy variable)	0.49
Scenic_val	Scenic beauty very important	0.69 (dummy variable)	0.46

The first step in the analysis was to estimate a bivariate probit model explaining *WTP* for protecting part and all remaining spruce-fir forests in the Southern Appalachian Mountains. Results are shown in table 17.3. As can be seen, logarithm of the offer amount is negative and statistically significant at the 0.01 level in both equations. Because offer amounts are varied across individuals, the variation in binary responses conveys information about the variance of the equation error (σ_i^2). Cameron and James (1987) show that the coefficient on the bid amount is a point estimate of $1/\sigma_i$. Taking the antilog

Table 17.3. Bivariate probit *WTP* results (N = 205)

Equation[a]	Constant	Ln_bid	Ln_inc	Enviro	Rec_val	Scenic_val
WTP part	-2.67*	-0.57***	0.35**	0.50**	0.53**	0.41
	(1.62)	(0.10)	(0.16)	(0.23)	(0.25)	(0.29)
WTP all	-1.97	-0.45***	0.28*	0.61***	0.08	0.56**
	(1.73)	(0.08)	(0.16)	(0.23)	(0.24)	(0.25)

* = significant at the10% level, ** = significant at the 5% level, *** = significant at the 1% level. Standard errors in parentheses.

[a] ρ=0.04(0.04)***

of the inverse of the parameter estimates on ln_bid in each equation, we compute $\sigma_1 = 5.80$ and $\sigma_2 = 9.35$. This result indicates that the standard error of the second equation (for protecting all remaining spruce-fir forests in the Southern Appalachian Mountains) is larger than the standard error of the first equation (for protecting spruce-fir forests along roads and trails only). Apparently, responses to the second *WTP* question contain more statistical noise than responses to the first equation.[5] Differences in error distributions for the two equations support the rationale for using median *WTP* values for hypothesis tests rather than mean *WTP* values.[6] We also note that the parameter estimate for the correlation coefficient is highly significant and close to one, justifying the use of the bivariate probit model.

Table 17.4 shows the median *WTP* estimates and the 95% confidence intervals computed using random draws from the multivariate distribution of the bivariate probit parameter estimates. In B = 1000 draws, median *WTP* for protecting part of the spruce-fir ecosystem along roads and trails always exceeded zero. Consequently, we conclude that people are willing to pay a positive amount to protect at least part of the forest ecosystem at risk.

Table 17.4. Empirical median *WTP* distributions using B = 1000 random draws

Protection level	Lower bound (0.05)	Median	Upper bound (0.05)
Roads and trails (WTP_a)	\$11.81	\$18.17	\$24.84
All remaining (WTP_{a+b})	\$18.02	\$28.49	\$40.96

Table 17.5 presents the results of the three bootstrap hypothesis tests described in section 5. Results indicate that incremental *WTP* for forest ecosystem protection is positive (\$28.49 ≠ \$18.17) and that incremental *WTP* increases at a decreasing rate (\$28.49 – 18.17 = \$10.32 < \$18.17). Consequently, we conclude that preferences for forest ecosystem protection, as obtained in this study, are well-behaved and are consistent with economic theory.

We note that the bootstrap hypothesis testing procedure described here is preferable to the non-overlapping confidence interval criterion used in an earlier treatment of the problem (Park et al. 1991). Under that criterion, the null hypothesis of no significant difference is rejected if the $(1 - \alpha)$ confidence intervals for *WTP* do not overlap. As pointed out by Poe et al. (1994), the actual significance level is higher than the significance level indicated by the test. This is consistent with our results.[7]

Table 17.5. Bootstrap hypothesis test results using the percentile method (B = 1000)

Null hypothesis	ASL%	Result
$H_0^1 : WTP_a = 0$	0.000	Reject H_0; Accept H_a
$H_0^2 : WTP_a = WTP_{a+b}$	0.009	Reject H_0; Accept H_a
$H_0^3 : (WTP_{a+b} - WTP_a)/b = (WTP_a/a)$	0.001	Reject H_0; Accept H_a

Finally, we report results for the value components of *WTP* (table 17.6). We recognize that there is debate in the literature about the cognitive ability of individuals to decompose total value in this way. However, we found that people allocated the greatest proportion of *WTP* to existence value. These results suggest that non-use values are an important component of total value for protection of this forest ecosystem.

Table 17.6. Value components of *WTP*

Type of value	Proportion of *WTP*	Component value
Use	0.13	$3.70
Bequest	0.30	$8.55
Existence	0.57	$16.24
Total	1	$28.49

7. CONCLUSION

Full and accurate assessment of forest values is essential for appraising projects and policies affecting the use of forests. Under-valuation of forest ecosystems can bias land use policies in directions that are not consistent with maximizing economic welfare. By improving the understanding of the economic importance of the structure, health, and extent of forest ecosystems, more informed forest policy and management decisions can be made.

The multiple outputs of forest ecosystems make their economic valuation challenging. This is particularly true when there are significant passive use values associated with protecting or restoring forest ecosystems. Contingent valuation is part of the tool kit available to forest resource economists. It allows a holistic approach to valuing the complex environmental good that a forest ecosystem represents.

A variety of studies using contingent valuation to value forest ecosystems have been conducted. The applications have included changes in forest quality due to insect infestations and air pollution, protection of existing ecosystems, and forest restoration. The studies show consistent support for the hypothesis that protection and restoration of forest ecosystems is an economic good that people are willing to pay for. Our own application to spruce-fir ecosystems confirmed this result and showed that consumer preferences regarding forest ecosystems were well-behaved and consistent with the constructs of economic theory. Thus, estimated *WTP* values can be used in cost/benefit assessments of forest ecosystem protection programs. These results were robust despite the fact that when *WTP* was decomposed, we found that existence value accounted for the greatest proportion of reported forest value.

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[1] While some analysts have used CVM to value individual elements of forest ecosystems, e.g. carbon sequestration or endangered species habitat, the focus of this chapter is on entire ecosystems.

[2] Johansson (1987:11) states that "A utility function $U(x)$ is *well-behaved* if (i) it is continuous where finite on X, (ii) it is increasing (and $\partial U(x)/\partial x_i > 0$ for all i), (iii) it is strictly quasi-concave on X, and (iv) it generates at least twice continuously differentiable demand functions."

[3] For each parameter in the estimated CVM model, random draws are made from a multivariate normal distribution with mean values set equal to the vector of parameter estimates and distribution set equal to the estimated variance-covariance matrix. Given the bootstrap parameter vector, median *WTP* is computed and stored. Computing and storing B bootstrap replications of median *WTP* yields a bootstrap distribution of the median for each equation. Sorting the median *WTP* bootstrap distribution allows confidence intervals to be established, and hypothesis tests can be constructed.

[4] A similar method to test for difference in mean *WTP* for nonindependent dichotomous choice responses was used by Poe et al. (1997).

[5] This result has also been observed using the double-bounded dichotomous choice format (Cameron and Quiggin 1994). In addition, this result is consistent with rank-order studies that indicate cognitive burden, and therefore respondent fatigue, increases with increasing rank (see chapter 18).

[6] It may be recalled that, in the log-linear specification, estimates of mean *WTP* are influenced by equation error. If mean values were used in the current application, estimates of mean *WTP* would be inflated for the second equation relative to the first. This would, in turn, affect the efficacy of the hypothesis test concerning whether *WTP* is the same in the two equations.

[7] A review of table 17.4 shows that the 95% confidence intervals overlap for the two distributions. However, the bootstrap hypothesis procedure shows that median *WTP* values are statistically different at the 1% significance level.